NCCARF
National
Climate Change Adaptation
Research Facility

Murdoch
UNIVERSITY

Australian Government
National Water Commission

Novel methods for managing freshwater refuges against climate change in southern Australia

Supporting document 2: Riparian replanting for temperature control in streams

Barbara Cook, Paul Close, Melanie Stock and Peter M. Davies

NOVEL METHODS FOR MANAGING FRESHWATER REFUGES AGAINST CLIMATE CHANGE IN SOUTHERN AUSTRALIA

SUPPORTING DOCUMENT 2:
Riparian replanting for temperature control in streams

AUTHORS

Barbara Cook – The University of Western Australia
Paul Close – The University of Western Australia
Melanie Stock – The University of Western Australia
Peter Davies – The University of Western Australia

NCCARF
National Climate Change Adaptation Research Facility

Australian Government
National Water Commission

The University of Western Australia

Published by the National Climate Change Adaptation Research Facility 2013

ISBN: 978-1-925039-52-8   NCCARF Publication 81/13

Please cite this report as:

Cook, BA, Close, PG, Stock, M & Davies, PM 2013, *Novel methods for managing freshwater refuges against climate change in southern Australia. Supporting Document 2: Riparian replanting for temperature control in streams*, National Climate Change Adaptation Research Facility, Gold Coast, 37 pp.

Acknowledgement

This work was carried out with financial support from the Australian Government (through the Department of Climate Change and Energy Efficiency and the National Water Commission) and the National Climate Change Adaptation Research Facility (NCCARF).

The role of NCCARF is to lead the research community in a national interdisciplinary effort to generate the information needed by decision-makers in government, business and in vulnerable sectors and communities to manage the risk of climate change impacts.

We thank the Department of Agriculture and Food, Western Australia for laboratory facilities and The University of Western Australia, Centre of Excellence in Natural Resource Management for administrative support. Peter Cook assisted with experimental determination of upper thermal tolerance of aquatic insects. We thank Belinda Robson (Murdoch University) for managing the project, of which this report is a component.

Table of contents

List of Figures

List of Tables

Abstract

Southern Australia is becoming warmer and drier as climate change progresses. This creates particular threats for freshwater ecosystems that are dependent on the presence of water for their existence. This project focused on riparian vegetation and its ability to mediate water temperature by reducing input of solar radiation through shading.

Unfortunately, resilience of freshwater ecosystems to predicted thermal shifts associated with climate change has been reduced by the widespread removal or degradation of riparian vegetation, characteristic of many temperate Australian streams. Many stream species in southern Australia are intolerant of elevated water temperatures, meaning that the maintenance of cool water refuges is critical to their sustainability, and consequently ecosystem health. *In situ* restoration of rivers and streams is a practical response to climate change and restoration efforts are prioritising riparian revegetation particularly in areas of current or predicted climate change.

To assist in the determination of optimal shading regimes for refuges using riparian plantings, this study (i) established both species-specific tolerances and community-level thresholds of concern using existing experimental data as well as relationships between species distribution and associated environmental data, (ii) developed the scenario testing capacity of the SimpSTREAMLINE model approach for developing a riparian replanting strategy that will provide relief from high temperatures, and (iii) tested this approach for selected case studies.

Executive Summary

Southern Australia is becoming warmer and drier as climate change progresses. This creates particular threats for freshwater ecosystems that are dependent on the presence of water for their existence. A broad project, established under the Australian Government Department of Climate Change and Energy Efficiency and the National Climate Change Adaptation Research Facility (NCCARF) program (Robson et al., 2013), aimed to develop and evaluate methods to enhance the role, function and resilience of refuges for freshwater biodiversity in southern Australia. This project focused on the assessment of novel methods to maintain the physical conditions in refuges within ranges tolerable for species and to maintain connectivity that allows species to retreat to, and expand from, refuges.

This broader project evaluated four novel methods:

Sub-project 1: Cold-water releases (shandying) for enhancing the resilience of riverine species (Cummings et al., 2013).
Sub-project 2: Riparian replanting for temperature control in streams (this study).
Sub-project 3: Anthropogenic habitats as freshwater refuges (Chester et al., 2013).
Sub-project 4: Modifying small barriers to improve connectivity in rivers (Beatty et al., 2013).

Key findings of these four sub-programs have been provided in separate technical reports (see references above) and summarised by Robson et al. (2013). This report provides key outcomes from Sub-project 2: *Riparian replanting for temperature control in streams*.

Riparian vegetation mediates water temperature by reducing input of solar radiation through shading (Rutherford et al., 2004). Unfortunately, resilience of these systems to predicted thermal shifts associated with climate change (Hennessy et al., 2007; Davies, 2010) has been reduced by the widespread removal or degradation of riparian vegetation, characteristic of many temperate Australian streams (see; Bunn et al., 1999; Davies et al., 2004; Armstrong et al., 2005).

Many stream species in southern Australia are intolerant of elevated water temperatures (Davies, 2010), meaning that the maintenance of cool water refuges is critical to their sustainability, and consequently ecosystem health. *In situ* restoration of rivers and streams is a practical response to climate change and restoration efforts are prioritising riparian revegetation particularly in areas of current or predicted climate change (see Bernhardt et al., 2005; Price et al., 2008; Catford et al., 2012).

Modelling approaches such as SimpSTREAMLINE (Rutherford et al., 1997; Davies et al., 2004) and others (Theurer et al., 1985; McBride et al., 1993) can be used to identify priority areas and determine the extent of riparian cover and length of rehabilitation in catchments for riparian replanting to restore or maintain stream temperatures within the thermal tolerance of keystone species (e.g. Rutherford et al., 1997; 2004). Such applications rely on thermal tolerance data for stream invertebrates; information that is currently lacking for most Australian stream taxa (*e.g.* McKie et al., 2004).

To assist in the determination of optimal shading regimes for refuges using riparian plantings, this study (i) established both species-specific tolerances and community-level thresholds of concern using existing experimental data as well as relationships between species distribution and associated environmental data, (ii) developed the scenario testing capacity of the SimpSTREAMLINE model approach adopted by

Davies et al. (2004) for developing a riparian replanting strategy that will provide relief from high temperatures, and (iii) tested this approach for selected case studies.

The key findings of this research were that:

- Experimentally derived Upper Thermal Tolerance (UTT) for selected Australian aquatic invertebrate taxa were similar to that of species tested elsewhere.
- Mean UTT (based on relevant literature and LT_{50} experiments) ranged from 22.3°C for Ephemeroptera to 43.4°C for Coleoptera.
- Mean UTT for both Coleoptera and Odonata (41.9°C) were significantly higher than those for all the other groups (22.3 –31.5°C) with the exception of Planaria. The mean UTT value of 22.3°C for Ephemeroptera was significantly lower than Decapoda (29.6°C), Trichoptera (30.1°C) and Mollusca (31.5°C). For three insect orders tested, eurytherms had significantly higher UTT values than stenotherms.
- Estimates of UTTs for aquatic insects based on Maximum Field Distribution (MFD) temperatures ranged from 18.4°C in the Ephemeroptera and Plecoptera to 22.2°C in the Coleoptera with values for the Ephemeroptera and Plecoptera significantly lower than for the Hemiptera and Coleoptera.
- Mean MFD temperatures for molluscs ranged from 16.1°C in the Hydrobiidae to 26.5°C in the Pomatiopsidae.
- Mean MFD temperatures were similar across the five major crustacean groups.
- A significant relationship between experimentally and field derived UTTs suggested that MFD temperature values could be useful in ascribing temperature sensitivity of a range of aquatic invertebrate taxa.
- The relative proportion of temperature-‘sensitive’, ‘tolerant’ and ‘very tolerant’ taxa changed among multiple pairs of shaded and unshaded sites, suggesting that UTT data for stream invertebrates can be used to set biodiversity targets for stream restoration aimed at temperature control.
- A five-step approach was developed to set biodiversity targets for stream restoration aimed at temperature control. This approach integrates outputs from modeling approaches, such as SimpSTREAMLINE, that predict the extent of cover and length of rehabilitation required to restore or maintain stream temperatures with the thermal tolerance of keystone species. This facilitates the establishment of adaptive riparian replanting strategies capable of providing refuges from otherwise high water temperatures.
- A Microsoft Access database was developed as a tool to set biodiversity targets and assess the biodiversity outcomes of proposed restoration activities.
- This approach could be used across southern Australia to assess whether planned restoration activities, aimed at mitigating against rising water temperatures, will provide for biodiversity outcomes.

1. PROJECT BACKGROUND AND INTRODUCTION

1.1 Novel methods for managing freshwater refuges against climate change in southern Australia.

Southern Australia is becoming warmer and drier as climate change progresses. This creates particular threats for freshwater ecosystems that are dependent on the presence of water for their existence. Much of the freshwater biota in southern Australia is comprised of, or derived from, cool stenotherms (cool-water temperate species; Davies 2010; Robson et al. 2012) and is therefore likely to be more sensitive to increased temperatures and more frequent and prolonged drying than the fauna of more arid areas.

A broad project, established under the Australian Government Department of Climate Change and Energy Efficiency and the National Climate Change Adaptation Research Facility (NCCARF) program (Robson et al., 2013), aimed to develop and evaluate methods to enhance the role, function and resilience of refuges for freshwater biodiversity in southern Australia. This project focused on the assessment of novel methods to maintain the physical conditions in refuges within ranges tolerable for species and to maintain connectivity that allows species to retreat to, and expand from, refuges. This broader project evaluated four novel methods:

Sub-project 1: Cold-water releases (shandying) for enhancing the resilience of riverine species (Cummings et al., 2013).
Sub-project 2: Riparian replanting for temperature control in streams (this study).
Sub-project 3: Anthropogenic habitats as freshwater refuges (Chester et al., 2013).
Sub-project 4: Modifying small barriers to improve connectivity in rivers (Beatty et al., 2013).

Key Findings of these four sub-programs have been provided in separate technical reports (see references above) and summarised by Robson et al. (2013). This report provides key outcomes from Sub-project 2: *Riparian replanting for temperature control in streams*.

1.2 Riparian replanting for temperature control in streams.

In small to mid-sized streams, riparian vegetation mediates water temperature by reducing input of solar radiation through shading (Rutherford et al., 2004). However, riparian zones are also often sites of intense disturbance and clearing, particularly in agricultural landscapes (Bunn et al., 1999). Many streams in southern Australia have had their riparian vegetation removed or severely altered, and thus these streams are no longer buffered from temperature extremes (see Davies et al., 2004). Based on a national assessment of river condition, Norris et al. (2001) identified that over 85% of assessed river reaches were highly modified due to catchment disturbance and, that compared with northern Australia, those reaches in New South Wales, South Australia and Western Australia had the greatest proportion of modified reaches. Around 47% of those rivers assessed during this audit were affected by changes to the riparian vegetation. Consequently, the resilience of these systems to additional thermal shifts associated with climate change is expected to be reduced. Southwestern Australia has already undergone significant climate change with drying and warming (CSIRO, 2007; Davies, 2010). A further 0.2ºC increase per decade is predicted for the next 30 years, resulting in a 2ºC increase by 2050 (Hennessy et al., 2007; Davies, 2010).

Stream invertebrates are a highly diverse component of aquatic communities and an important food web link between primary sources of carbon (*e.g.* detritus, algae) and higher-order consumers including fish (Bunn et al., 1999). Due to their sensitivity to changes in flow (*e.g.* Poff et al., 1997; Horwitz et al., 2008) and water chemistry (*e.g.* Bunn & Davies, 1992), they have been used in a range of biomonitoring programs (Bunn & Davies, 2000). Temperature is considered an important control on stream communities (De Deckker, 1986).

Water temperatures may affect aquatic biota directly (e.g thermal tolerance), or indirectly (e.g. through its influence on dissolved oxygen concentrations). Instream water temperatures reduce the solubility of oxygen and increase rates of ecosystem respiration, thus reducing the availability of dissolved oxygen (DO) in water (Bunn & Davies, 1992; Horne & Goldman, 1994; Bunn et al., 1999). Aquatic biota may also respond directly to the entire thermal regime, including absolute temperatures, diel and seasonal amplitudes and rates of change (Ward & Stanford, 1982). Response of aquatic biota to thermal shifts may be driven by either sublethal effects, or in cases where thermal tolerance is exceeded, by direct lethal effects (Ward & Stanford, 1982).

Many stream species in southern Australia have Gondwanic origins (Bunn & Davies 1990) and are considered cold stenotherms, intolerant of elevated water temperatures (Davies, 2010). The maintenance of cool water refuges is therefore critical to the sustainability of these aquatic invertebrates. Indeed, it is widely believed that water temperature is a major factor restricting Gondwanan species in southern Australia (*e.g.* Bunn & Davies, 1990; McKie et al., 2004). The importance of maintaining suitable temperature regimes is also relevant to eurytherm species, typical of inland lowland waterways (e.g the Murray darling basin). Although the upper thermal tolerance of these species is likely to be higher than that for stenotherms, maintenance of appropriate water temperatures will be critical to their survival

In situ restoration of rivers and streams is a practical response to climate change in the short to medium term. This may be partly achieved through better reserve design (Dunlop & Brown, 2008), maintenance of off-reserve biodiversity, reducing other threats (*e.g.* secondary salinization; Horwitz et al., 2008) or through restoration that increases overall resilience (see Davies, 2010). Removal of riparian vegetation has been a widespread feature of Australian landscapes (Bunn et al., 1999; Armstrong et al., 2005;) and restoration efforts are prioritising riparian revegetation particularly in areas of current or predicted climate change (see Catford et al., 2012), both in Australia (e.g. Price et al., 2008) and elsewhere (see Bernhardt et al., 2005).

Indeed, evidence strongly supports the notion that stream temperatures, and particularly maximum temperatures, are significantly influenced by riparian shading (e.g. Beschta 1997; Johnson & Jones 2000; Johnson 2004; Rutherford et al. 2004). For example, Johnson (2004) successfully decreased maximum stream temperatures in an Oregon stream using artificial shading, and Rutherford et al. (2004) found that stream temperatures were high in the absence of riparian vegetation and lower in the presence of intact riparian vegetation (Figure 1) for a second-order stream near Albany in Western Australia. The control of water temperature through riparian shading is an area of restoration where target values can be set and consequently the amount of vegetation required to meet these targets can be specified (Davies et al., 2004).

The ability to predict characteristics of future ecosystems is crucial for environmental planning and the development of effective climate change adaptation strategies (Davies, 2010). Modelling studies have demonstrated that planting trees on stream banks can reduce daily maximum water temperatures (Theurer et al., 1985; McBride et al., 1993). In particular, the SimpSTREAMLINE model developed by Rutherford et al. (1997) and modified by Davies et al. (2004) was developed to predict daily temperature fluctuations in streams. When used in combination with digital elevation models for mapping solar radiation and maps showing distribution of stream vegetation, it can be used to identify priority areas in catchments where replanting will increase shading and decrease temperatures.

Figure 1. Examples of riparian conditions characteristic of unshaded (left) and shaded (right) streams in southwestern Australia.

These models have been applied to predict the extent of cover and length of rehabilitation required to restore or maintain stream temperatures within the thermal tolerance of keystone species (e.g. Rutherford et al., 1997; 2004). Such applications rely on thermal tolerance data for stream invertebrates. Although the thermal tolerances of aquatic invertebrates occurring in streams in USA (e.g. De Kozlowski & Bunting, 1981; Claussen & Walters, 1982), South Africa (Buchanan et al., 1988) and New Zealand (Quinn et al., 1994) have been determined, application of these models in Australia is not possible because the thermal tolerances of Australian stream invertebrates remain largely unknown (*e.g.* McKie et al., 2004).

To assist in the determination of optimal shading regimes for refuges using riparian plantings, this study (i) established both species-specific tolerances and community-level thresholds of concern using existing experimental data as well as relationships between species distribution and associated environmental data, (ii) developed the scenario testing capacity of the SimpSTREAMLINE model approach adopted by Davies et al. (2004) for developing a riparian replanting strategy that will provide relief from high temperatures for refuge biodiversity, and (iii) tested this approach for selected case studies.

1.3 Structure of the report

This report details key findings for Sub-project 2; *Riparian replanting for temperature control in streams*, and is structured by the following four, stand-alone chapters:

Chapter 1: General introduction including context and objectives of the broader NCCARF Refuge Project, background, introduction and aims for the Sub-project 2: *Riparian replanting for temperature control in streams*.
Chapter 2: Determination of upper thermal tolerances of aquatic invertebrates using experimental data.
Chapter 3: Determination of upper thermal tolerances of aquatic invertebrates using field data.
Chapter 4: Using upper thermal tolerances to set biodiversity targets for riparian restoration.

2. DETERMINING UPPER THERMAL TOLERANCES USING EXPERIMENTAL DATA

2.1 Objectives

While the capacity to maintain or reinstate cooler temperatures through riparian restoration has been demonstrated (Rutherford et al., 1997; 2004), the ability to predict ecological response, particularly change in assemblage structure, to those actions is limited by knowledge on the thermal tolerance of key taxonomic groups.

The primary aim of this Chapter is twofold. Firstly, we review available data on the upper thermal tolerance limits of aquatic invertebrates, relating these tolerances to taxonomic groups and acclimation temperatures. Secondly, we present the results of an investigation using standard 96h LT_{50} tests of the thermal tolerances of four key southwestern Australian taxa. This represents the first investigation of thermal tolerances of species from this region. Together, the data from the review and these experiments, will allow the formulation of temperature targets for riparian restoration both in Australia and world-wide.

2.2 Methods

2.2.1 Laboratory experiments

In order to facilitate the selection of a range of organisms with a wide variation in upper thermal tolerances for the LT_{50} experiments, macroinvertebrate community structure was initially compared among eight sites along Marbellup Brook in the Torbay catchment, Western Australia; four 'shaded' sites with intact riparian vegetation, and four 'unshaded' sites devoid of riparian trees. At each site, macroinvertebrates were collected by sweeping a 250-µm mesh net over $10 m^2$ of stream bed, disturbing the top few centimetres of substrate. Leaves, twigs and other debris were rinsed and discarded, and animals were returned to the laboratory where they were identified to family level. Community structure among the sites was compared using the software package PRIMER v5 (Clarke, 1993). After calculating similarities between every pair of the eight samples using the Bray-Curtis coefficient, samples were clustered using the UPGMA algorithm. Significant differences among assemblages were tested using ANOSIM.

Species that primarily accounted for the observed assemblage differences were identified by the SIMPER routine, such that the overall percentage contribution each species made to the average dissimilarity between the two groups was calculated, and species were listed in decreasing order of their importance in discriminating the two sets of samples. Based on the results of these analyses and specimen availability, four species were selected for the LT_{50} experiments: the caddisfly *Cheumatopsyche modica* (family Hydropsychidae) and the mayflies *Offadens soror* (Baetidae) and *Nyungara bunni* (Leptophlebiidae), all 'typical' of shaded sites (but poorly represented or absent at unshaded sites), and the dragonfly *Austroaeschna anacantha* (Telephleiidae), found consistently at both shaded and unshaded sites. Three of these species (*N. bunni, A. anacantha* and *C. modica*) are endemic to southwestern Australia and are considered to be gondwanic relicts.

Individuals of the selected taxa were collected from Marbellup Brook using the same methodology described above before transfer to aerated buckets using wide-mouthed pipettes.

On return to the laboratory, animals were transferred to 200ml plastic containers (five individuals in each) containing pre-conditioned, filtered river water, and for each species, five replicate, aerated containers were placed in constant temperature water baths (Figure 2). The baths were initially set to 15°C, and the animals were acclimated for four days at this temperature. In previous investigations aquatic invertebrates have been acclimated for between three (Gaufin & Hern, 1971; De Kozolwski & Bunting, 1981; Moulton et al., 1993) and 12 days (Claussen & Walters, 1982). As is commonplace in investigations of this type, animals were not fed during experiments (e.g. Claussen & Walters, 1982; Buchanan et al., 1988). All experiments were conducted in a laboratory with a natural diurnal light regime (due to the presence of large, external windows) and baths subjected to treatments were randomly positioned. Thermal tolerance at one control (15°C) and five different test temperatures (25, 29, 33, 37 and 41°C) were assessed for the dragonfly and caddisfly species, and four test temperatures (21, 25, 29 and 33°C) for the two mayfly species. Acclimation temperatures were chosen to reflect likely environmental temperatures to which wild populations are exposed in both shaded and unshaded streams.

To avoid thermal shock, temperatures in the individual water baths were manually raised by 2-3°C per hour until the desired experimental temperatures were maintained to within 0.5°C. Submersible pumps in each bath ensured that the temperature was evenly distributed, and temperatures in each container were constantly monitored to ensure that they remained at the target level. Survival was recorded at three endpoints (24, 48 and 96h) after the target temperature was reached. The temperature at which 50% of the organisms died (LT_{50} values) and 95% fiducial limits were calculated for each time period by probit analysis following the EPA flowchart procedure outlined in ToxCalc, a toxicity data analysis and database software package (Tidepool Scientific Software and Micheal A. Ives, 1994-1996). A trimmed Spearman-Karber analysis was used to estimate LT_{50} values where data did not fit the probit model.

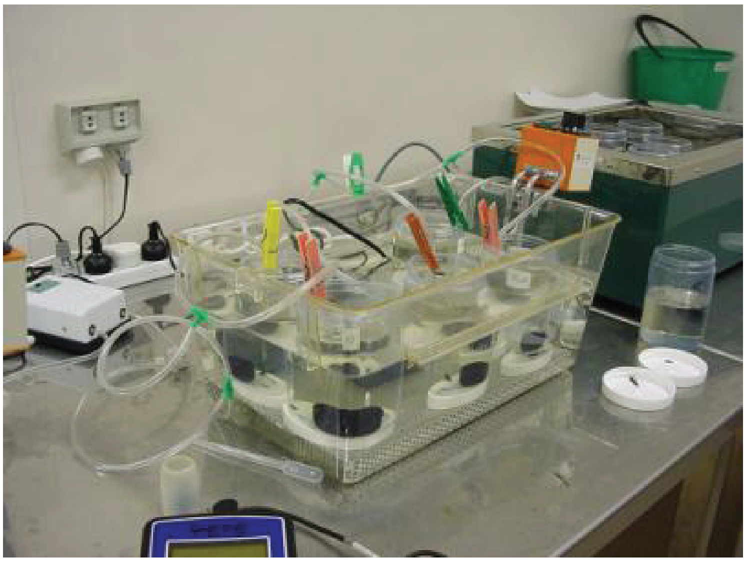

Figure 2. Experimental set-up for determination of LT50, showing replicate experimental units in constant temperature water bath.

2.2.2 Review of upper thermal tolerance levels

Traditionally, upper thermal tolerance has been determined in the laboratory using either time to death at constant test temperatures (the lethal temperature, or LT method), or the critical thermal maximum (CTM_{ax}) method, which involves increasing test temperatures until an end point is reached. Of these two approaches, the use of the LT methodology has been favoured for invertebrates, although there has been a general trend across all major faunal groups to move from using LT methods to CTM_{ax} methods (Lutterschmidt & Hutchison, 1997). Only 22% of invertebrate studies reviewed by Lutterschmidt & Hutchison (1997) used CTM_{ax} methods, and these authors suggested that this might be due to the fact that the onset of muscular spasms (common endpoint in CTMax studies) is difficult to observe in many invertebrates. Working on fish, Kilgour & McCauley (1986) constructed a heuristic model, which showed that these two experimental procedures are closely related, and that data from either can provide a reasonable prediction of results from the other approach. Similarly, in an investigation of upper thermal temperatures for dragonfly nymphs, Garten and Gentry (1976) found that LT estimates were significantly correlated with CTM_{ax} values for the species examined.

A comprehensive search revealed a limited literature on upper thermal tolerance of aquatic invertebrates, with 19 papers on thermal tolerance of aquatic invertebrates published in the period 1968 to 2008 (see Table 2). For each study, we noted the procedures and methods used (LT method or CTM_{ax}), species tested and geographical location of specimen collection. We also recorded acclimation temperatures as a species' thermal history immediately prior to testing is known to influence thermal tolerances (Lutterschmidt & Hutchison, 1997). A total of 11 of the 19 studies assessed (58%) used the LT method; six studies used CTMax (32%) and two studies used both methods.

Significant differences in UTT among broad taxonomic groups (usually order or class level), and between acclimation temperature categories (acclimated at temperatures below 15^{o}C or at temperatures of 15^{o}C or above) and stenotherm and eurytherm species within the major taxonomic groups were detected using analysis of variance (ANOVA) and Tukey's multiple comparison tests. Upper thermal tolerance levels were compared amongst major groups of macroinvertebrates, and amongst families within selected groups using box-and-whisker diagrams (Tukey, 1977). Species were classified as stenotherms if they were either known to occur naturally in cold streams, or were known to emerge in early spring prior to elevated summer water temperatures. Those species classified eurytherms in our study were either known to inhabit warmer slow moving streams, or had longer life cycles emerging after exposure to elevated summer water temperatures.

2.3 Results

2.3.1 Laboratory experiments

Control mortality in laboratory procedures was low and varied from 0-4% in all experiments. Dragonflies were the least sensitive to high temperatures with a LT_{50} value of 33.5°C after 96h exposure (Table 1). Few deaths were recorded at either 25°C or 29°C after 96h (Figure 3). Mortality increased at 33°C, and at 37°C, 48% of animals had died after 24h, and by 48h, all remaining animals had died. At 41°C, all animals died within 24h. Caddisflies were more sensitive to high water temperatures than dragonflies. Although few animals died at 25°C and 29°C, significant mortality (96%) occurred at 33°C after only 24h. An LT_{50} value of 30.7°C after 96h exposure was calculated for this species. The two species of mayflies tested were the most sensitive to high water temperatures, with LT_{50} values of 20.5°C estimated for the baetid *Offadens soror*, and 21.9°C for the leptophlebiid *Nyungara bunni* after 96h exposure.

Table 1. LT50 values with associated 95% fiducial limits (where given by probit analysis) and 95% confidence limits (where given by Spearman-Karber analysis) for four southwestern Australian species of aquatic invertebrates at 24, 48 and 96h. a = estimated using probit analysis. b = estimated using Spearman-Karber analysis.

Taxa	24 hour		48 hour		96 hour	
	LT_{50}	95% limits	LT_{50}	95% limits	LT_{50}	95% limits
Offadens soror	26.6	25.5-27.6	23.7	22.7-24.7	20.5	-
Nyungara bunni	-	-	-	-	21.9	-
Austroaeschna anacantha	36.8^{a}	35.9-37.7	34.3^{b}	33.6-34.9	33.9^{b}	33.1-34.6
Cheumatopsyche modica	30.7	29.7-31.6	30.6	29.6-31.5	30.7	29.5-31.6

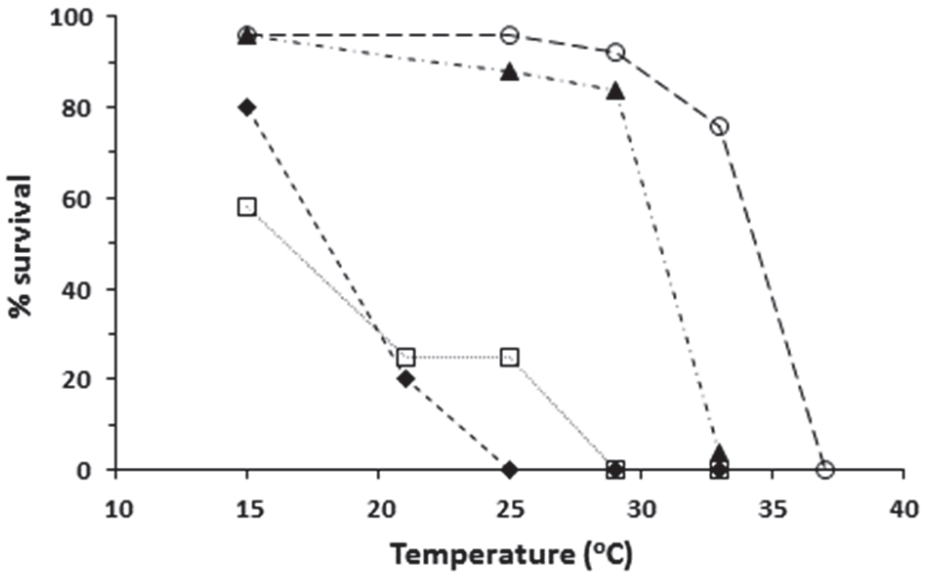

Figure 3. Plot of temperature versus % survival after 96h exposure for four species of aquatic invertebrates from southwestern Australia. Symbols are as follows: diamond = *Offadens soror*, square = *Nyungara bunni*, triangle = *Cheumatopsyche modica*, circle = *Austroaeschna anacantha*.

2.3.2 Review of upper thermal tolerance levels

Review of published literature (including laboratory results from this study) revealed upper thermal tolerance (UTT) data for 81 species in 40 invertebrate families (or subfamilies) (Table 2). Mean upper thermal tolerance among the major macroinvertebrate taxonomic groups, tested in laboratory studies worldwide (including results from this study) ranged from 22.3°C for Ephemeroptera (mayflies) to 43.4°C for Coleoptera (beetles) (Fig. 4; Table 3). Mean thermal tolerance levels for Coleoptera (43.4°C) and Odonata (41.9°C) were similar (ANOVA, $p > 0.05$), but significantly higher than mean values for all the other groups assessed (range from 22.3°C to 31.5°C) with the exception of Planaria (ANOVA, $p < 0.05$). The mean value of 22.3°C for Ephemeroptera was significantly lower than for Decapoda (29.6°C), Trichoptera (30.1°C) and Mollusca (31.5°C) (ANOVA, $p < 0.05$).

Table 2. Mean upper thermal tolerances (UTT) for families of aquatic macroinvertebrates as determined in a review of laboratory experiments using either lethal temperature (LT) or critical thermal maximum (CTM) approaches. Literature sources from which the data were extracted are indicated. Families in bold include LT50 data from this study. S.E. = standard error.

Group	Family/ subfamily	UTT (°C)	S.E.	Localities	Source
Planaria	Dugesiidae	32.2	0.3	USA	Claussen & Walters, 1982
Oligochaeta	Lumbriculidae	26.7	-	New Zealand	Quinn et al., 1994
Mollusca	Hydrobiidae	31.8	0.4	New Zealand	Winterbourn, 1969; Cox & Rutherford, 2000
	Sphaeridae	30.5	-	New Zealand	Quinn et al., 1994
Amphipoda	Eusiridae	24.1	-	New Zealand	Quinn et al., 1994
	Gammaridae	14.6	-	USA	Gaufin & Hern, 1971
	Paramelitidae	34.1	-	South Africa	Buchanan et al., 1988
Decapoda	Astacidae	30.6	0.9	New Zealand, Japan	Simons, 1984; Nakata et al., 2002
	Atyidae	25.8	0.1	New Zealand	Davenport & Simons, 1985; Quinn et al., 1994
	Cambaridae	32.3	2.7	USA, Japan	Claussen, 1980; Nakata et al., 2002
Ephemeroptera	**Baetidae**	20.1	-	Australia	Present study
	Ephemerellidae	20.4	1.7	USA	Nebeker & Lemke, 1968; Gaufin & Hern, 1971; De Kozlowski & Bunting, 1981
	Ephemeridae	26.6	-	USA	Gaufin & Hern, 1971
	Heptageniidae	23.0	5.9	USA	Nebeker & Lemke, 1968; Gaufin & Hern, 1971

Group	Family/ subfamily	UTT (°C)	S. E.	Localities	Source
	Leptophlebiidae	23.1	0.5	Australia, New Zealand	Quinn et al., 1994; Cox & Rutherford, 2000; Present study
Odonata	**Aeshnidae**	33.2	0.7	Australia, USA	Nebeker & Lemke, 1968; Present study
	Corduliidae	41.1	1.1	USA	Garten & Gentry, 1976
	Gomphidae	33.0	-	USA	Nebeker & Lemke, 1968
	Libellulidae	43.7	0.5	USA	Martin & Gentry, 1974; Garten & Gentry, 1976
	Macromiidae	41.0	2.2	USA	Garten & Gentry, 1976
Plecoptera	Capmidae	23.0	-	USA	Nebeker & Lemke, 1968
	Gripopterygidae	25.5	-	New Zealand	Quinn et al., 1994
	Nemouridae	31.5	-	USA	Ernst et al., 1984
	Perlidae	31.7	0.9	USA	Nebeker & Lemke, 1968; Heiman & Knight, 1972; Ernst et al. 1984
	Perlodidae	24.1	4.9	USA	Nebeker & Lemke, 1968; Gaufin & Hern, 1971; Ernst et al., 1984
	Pteronarcyidae	27.0	1.5	USA	Nebeker & Lemke, 1968; Gaufin & Hern, 1971
	Taeniopterygidae	21.0	-	USA	Nebeker & Lemke, 1968
Trichoptera	Brachycentridae	30.5	1.2	USA	Nebeker & Lemke, 1968; Gaufin & Hern, 1971; De Kozlowski & Bunting, 1981
	Conoesucidae	28.7	3.7	New Zealand	Quinn et al., 1994

Group	Family/ subfamily	UTT (^oC)	S.E.	Localities	Source
	Hydropsychidae	30.4	1.7	Australia, New Zealand, USA	Gaufin & Hern, 1971; De Kozlowski & Bunting, 1981; Moulton et al., 1993; Quinn et al., 1994; Present study
	Limnephilidae	26.1	1.3	USA	Gaufin & Hern, 1971;
	Philopotamidae	33.8	1.5	USA	Moulton et al., 1993
	Uenoidae	25.9	-	USA	Gaufin & Hern, 1971
Diptera	Athericidae	32.2	0.2	USA	Nebeker & Lemke, 1968; Gaufin & Hern, 1971
	Chironominae	24.1	-	Australia	McKie et al., 2004
	Orthocladiinae	27.2	2.4	Australia	McKie et al., 2004
	Simuliidae	25.1	-	USA	Gaufin & Hern, 1971
	Tanypodinae	25.3	3.5	Australia	McKie et al., 2004
Coleoptera	Dytiscidae	44.8	0.4	UK	Calosi et al., 2008
	Elmidae	32.6	-	New Zealand	Quinn et al., 2004

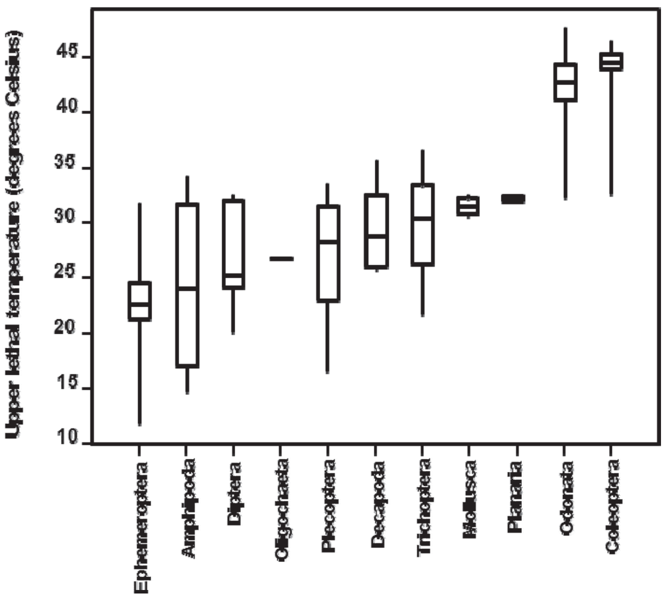

Figure 4. Plot of upper lethal temperatures (UTT) for major taxonomic groups of aquatic invertebrates. Boxes span the interquartile range of values, horizontal line indicates the median, vertical whiskers extending beyond the boxes represent minimum and maximum values.

Table 3. Mean upper thermal tolerance levels (UTT) as determined in a review of laboratory experiments using either lethal temperature (LT) or critical thermal maximum (CTM) approaches for selected macroinvertebrate groups. n = sample size, S.E. = standard error. Lower case letters indicate results of statistical tests for significant differences. Means that share the same letter are not significantly different, means that have different letters are significantly different.

Taxonomic Group	Mean UTT (^oC)	n	S.E.	Minimum-maximum (^oC)
Planaria	32.2^{abc}	2	0.3	31.9-32.4
Oligochaeta	26.7^{ab}	1	-	-
Mollusca	31.5^{b}	4	0.4	30.5-32.4
Amphipoda	24.3^{ab}	3	5.6	14.6-34.1
Decapoda	29.6^{b}	9	1.3	25.7-35.6
Ephemeroptera	22.3^{a}	13	1.4	11.7-31.8
Odonata	41.9^{c}	27	0.7	32.5-47.6
Plecoptera	27.2^{ab}	14	1.4	16.5-33.4
Trichoptera	30.1^{b}	19	0.9	21.7-36.5
Diptera	27.2^{ab}	10	1.4	20.1-32.4
Coleoptera	43.4^{c}	9	1.4	32.6-46.4

Mean UTT values did not vary significantly between acclimation categories (those acclimated at temperatures below 15^{o}C versus those acclimated at 15^{o}C and above) for each of the orders Ephemeroptera, Plecoptera and Trichoptera (ANOVA, p ranging from 0.1 to 0.86). However, when mean UTT for stenotherms versus eurytherms were compared in these three insect orders, these were found to be significantly different at the 5% level in the Ephemeroptera and Trichoptera (Student t-test; $p < 0.05$), and marginally significantly different in the Plecoptera (p=0.07) (Table 4).

Table 4. Mean upper thermal tolerance levels (UTT) as determined in a global review of laboratory experiments using either lethal temperature (LT) or critical thermal maximum (CTM) approaches for species of Ephemeroptera, Plecoptera and Trichoptera. Stenotherms = species occurring in naturally cold streams, Eurytherms = species occurring in warmer streams with variable temperature regimes, n = sample size, S.E. = standard error, p-value = probability that means are significantly different.

Order	Group	Mean UTT (^oC)	S.E.	n	p-value
Ephemeroptera	Stenotherms	21.04	1.21	11	
	Eurytherms	29.2	2.60	2	0.02
Plecoptera	Stenotherms	24.74	2.22	7	
	Eurytherms	29.69	1.20	7	0.07
Trichoptera	Stenotherms	28.35	1.13	12	
	Eurytherms	33.06	0.95	7	0.01

2.4 Discussion

Early studies of the UTTs of aquatic invertebrates mainly centered in the USA due to a concern that heated water from steam-electric power generating facilities would have a detrimental effect on the biota (e.g. Gaufin & Hern, 1971). Since then, the literature on thermal tolerance has grown to include a variety of invertebrate taxa from a range of regions, presenting the opportunity for review of broad patterns in upper thermal tolerance of aquatic invertebrates. The results of our laboratory experiments and review of the literature confirm considerable taxonomic differences in ability to tolerate high water temperatures. Mayflies (Ephemeroptera) and Stoneflies (Plecoptera) were shown to be particularly sensitive (e.g. Ward & Stanford, 1982; Quinn et al., 1994), supporting the use of this group as a part of commonly-used EPT (Ephemeroptera, Plecoptera and Trichoptera) index for testing of ecological water quality.

These taxa contrast with the higher tolerance levels of beetles (Coleoptera), dragonflies (Odonata), and to a lesser extent, planarians. While thermal tolerance studies on dragonflies are limited, it does appear that these animals are able to tolerate higher temperatures than many other components of the freshwater fauna (e.g. Martin & Gentry, 1974). In the case of the planarians, Claussen and Walters (1982) suggested that the high thermal tolerances of these animals corresponded with their widespread and eurythermic distributions.

The Ephemeroptera is likely to be adversely affected by significant increases in stream temperatures, which might arise as a consequence of climate change (see Davies, 2010) and/or land use practices (see Horwitz et al., 2008). Although the order Ephemeroptera is relatively species poor in southwestern Australia (only 12 species) when compared to other parts of Australia (over 140 species), the fauna in the region is unique, with 58% of these species being endemic to southwestern Australia, and 83% being endemic to Western Australia (Davies & Stewart, submitted).

Our study clearly showed that mean upper thermal tolerances can differ within taxonomic groups. For example, mean upper thermal tolerances of eurytherms were significantly higher than that of stenotherms in Ephemeroptera, Plecoptera and Trichoptera. The stenotherm species included in our analysis were either restricted to cold headwater streams, or were known to emerge in early spring prior to the occurrence of elevated water temperatures that occur during summer. Eurytherm species were usually more widespread in distribution and had longer life cycles and thus were present in streams at elevated water temperatures. These observations are consistent with those of Calosi et al. (2008; 2010), who found that widespread European diving beetle taxa have significantly higher UTTs than restricted taxa.

The present study has provided the first estimates of upper thermal tolerance of southwestern Australian species. The data generated for four species (representing the insect orders Odonata, Trichoptera and Ephemeroptera) was in agreement with those obtained for other mayfly, dragonfly and caddisfly species from New Zealand (Quinn et al., 1994; Cox & Rutherford, 2000) and USA (Nebeker & Lemke, 1968; Gaufin & Hern, 1971; Martin & Gentry, 1974; Garten & Gentry, 1976; De Kowzlowski & Bunting, 1981). Published estimates of upper thermal tolerance of Australian species of aquatic invertebrates are very limited, with only the Chironomidae receiving attention (McKie et al., 2004). The fact that data obtained for Australian species is similar to values in the global literature enables the utilisation of this substantial body of additional information for predicting ecological responses of future warming of streams.

3. DETERMINING UPPER THERMAL TOLERANCES USING FIELD DATA

3.1 Objectives

Historically, estimation of upper thermal tolerances has been examined using two approaches, laboratory experimentation (e.g. LT_{50} experiments, see Chapter 1) and field studies. The advantage of these controlled laboratory experiments is that by exposing animals to known temperatures, a causal link can be established between temperature and the measured effect. However, laboratory experiments are usually conducted under artificial conditions, without taking into account other stressors that may be present in nature. In addition, our review of UTTs for stream invertebrates (Chapter 2) showed that globally, published data on upper thermal tolerance of stream invertebrates is limited, and exists for only about 80 species in 40 families. Studies of Australian species are even more limited, with only four species tested to date (present study).

The relative lack of laboratory-derived data on temperature tolerances for aquatic invertebrates means that UTTs need to be estimated based on field distributions. To date, there are no specific studies that report the maximum field temperature at which particular taxa have been collected in Australia. Although field studies of invertebrate presence/absence at sites of different temperatures do not establish a causal link between temperature and mortality, these studies do give some indication of the likely temperature tolerances of species. In addition, if a relationship between these two measures of UTT can be demonstrated, then either approach could be used to estimate temperature tolerances. While studies have been conducted to explore the relationship between laboratory- and field-derived salinity tolerances in Australia (Kefford et al. 2004; Horrigan et al. 2007), studies aimed at investigating the relationship between laboratory and field-derived temperature tolerances have yet to be undertaken.

In this chapter, we use existing distribution data for freshwater invertebrates from southwestern Australia to (i) estimate the UTTs of aquatic invertebrates based on maximum field temperatures, (ii) compare these temperatures derived from field studies to UTT values derived from laboratory studies, and (iii) compare the structure of invertebrate assemblages among stream sites to test the hypothesis that shaded sites would have significantly more temperature-sensitive taxa than unshaded sites.

3.2 Methods

3.2.1 Maximum field distribution temperatures

Existing data sourced from previous surveys were used in this study to determine the maximum temperature at which aquatic invertebrates have been recorded in nature in southwestern Australia, the so called maximum field distribution (MFD) (Kefford et al. 2004). Overall, a total of 14,950 records of species – temperature occurrences at 188 sites sampled as part of a study of the ecological values of South Coast rivers (Stewart, 2011), 230 sites from an investigation of the impacts of salinity on aquatic invertebrates in wetlands of southwestern Australia (Pinder et al. 2005), and additional sites sampled from the Blackwood River as part of a ongoing monitoring program (Pinder, DEC, unpublished data) were examined. Where more than one MFD was recorded for a family or higher taxon level, the mean of species-specific values were used to represent the family, or higher taxon level MFD.

Each taxon was assigned to one of three broad sensitivity categories (sensitive, tolerant and very tolerant) using percentiles derived from the distribution of MFD values. Analyses were restricted to insects, molluscs, crustaceans.

Mean MFD was compared among groups (orders in insects, families in molluscs and crustaceans) using ANOVA in the GenStat Statistical Software Package. Means were considered significantly different if p was < 0.05, and marginally significantly different if p was between 0.05 and 0.1.

3.2.2 Relationship between experimental and field data

For the comparison of laboratory and field derived upper thermal tolerance values, experimental temperature data for freshwater invertebrates from Chapter 1 were used. This Chapter reported LT_{50} values ranging from 14.6^{o}C (for amphipods in the family Gammaridae) to 44.8^{o}c (beetles in the family Dytiscidae). A total of 16 invertebrate families were available with both laboratory LT_{50} and MFD estimates for two or more species. Family level identification is commonly used in monitoring programs in Australia and has been used as the focus of analysis in the present study.

3.2.3 Scenario testing

To test the hypothesis that shaded streams would have a significantly higher proportion of temperature 'sensitive' taxa relative to nearby unshaded streams, the proportion of 'sensitive', 'tolerant' and 'very tolerant' taxa was compared between five pairs of sites in the Marbellup Brook catchment, Western Australia (Figure 5) sampled on two different occasions. Chi-square analyses were used to test for goodness of fit of observed and expected frequencies. The latter were calculated using the occurrence of the three categories at nearby shaded sites.

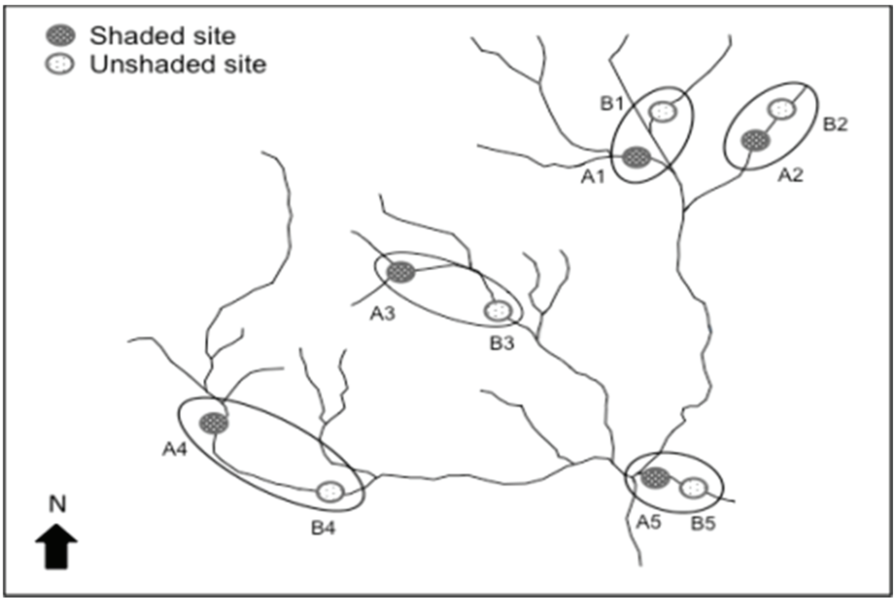

Figure 5. Five paired sites in the Marbellup Brook catchment, Western Australia (A = shaded and B = unshaded) sampled in 2006 and 2007 and used to assess differences in frequencies of temperature sensitivity categories ('sensitive', 'tolerant' and 'very tolerant') of invertebrates.

3.3 Results

3.3.1 Maximum field distribution temperatures

Table 5 summarises MFDs for families of insects, molluscs and crustaceans occurring in southwestern Australia. Mean MFD temperatures were estimated at both order and family levels for insects.

Table 5. Summary of mean maximum field distribution (MFD) temperatures for families of crustaceans, molluscs and insects in southwestern Australia. Min = minimum MFD recorded, Max = maximum MFD recorded, n = sample size, s.d. = standard deviation.

Family	Group	n	Mean	Min	Max	s.d.
Aeshnidae	Hexapoda	4	23.85	15.6	36.6	8.964
Ameiridae	Crustacea	6	19.63	15.3	26	3.565
Amphisopidae	Crustacea	2	19.51	16.6	22.42	4.115
Ancylidae	Mollusca	3	23.68	14.9	36.6	11.43
Artemiidae	Crustacea	1	29	29	29	
Athericidae	Hexapoda	1	13.2	13.2	13.2	
Atriplectididae	Hexapoda	1	16.38	16.38	16.38	
Austrocorduliidae	Hexapoda	1	23.05	23.05	23.05	
Baetidae	Hexapoda	5	18.71	15.7	25	3.623
Caenidae	Hexapoda	1	25	25	25	
Calanoidae	Crustacea	2	19.8	17	22.61	3.967
Candonidae	Crustacea	1	36.6	36.6	36.6	
Canthocamptidae	Crustacea	16	20.45	12	36.6	6.348
Carabidae	Hexapoda	1	29	29	29	
Ceinidae	Crustacea	1	15.5	15.5	15.5	
Centropagidae	Crustacea	18	22.24	15	36.6	6.127
Ceratopogonidae	Hexapoda	30	21.9	13	36.6	5.154
Chaoboridae	Hexapoda	1	17.9	17.9	17.9	
Chiltoniidae	Crustacea	1	30.8	30.8	30.8	
Chironomidae	Hexapoda	130	20.42	10.7	36.6	5.566
Chrysomelidae	Hexapoda	2	19.68	18.17	21.18	2.128
Chydoridae	Crustacea	58	21.48	7.5	36.6	6.757
Coenagrionidae	Hexapoda	9	21.19	13.85	27	5.149
Corduliidae	Hexapoda	5	21.38	17.7	23.19	2.397
Corixidae	Hexapoda	17	22.7	16.44	27	3.258
Corophiidae	Crustacea	1	16	16	16	
Culicidae	Hexapoda	30	19.21	14	27	3.352
Curculionidae	Hexapoda	2	23.85	19.5	28.2	6.152
Cyclopoidae	Crustacea	29	21.41	14	30.8	4.352
Cyprididae	Crustacea	78	23.54	7.5	36.6	6.876
Cypridopsidae	Crustacea	2	21	15	27	8.485
Cytherideidae	Crustacea	1	22	22	22	
Cyzicidae	Crustacea	3	19.87	14	25.6	5.801
Daphniidae	Crustacea	34	23.23	13.4	36.6	7.225

Family	Group	n	Mean	Min	Max	s.d.
Diosaccidae	Crustacea	4	18.77	14	23	4.45
Dolichopodidae	Hexapoda	3	26.73	25.2	29	2.003
Dytiscidae	Hexapoda	71	21.46	12	36.6	5.207
Ecnomidae	Hexapoda	10	17.64	12	26	4.306
Empididae	Hexapoda	1	23	23	23	
Ephydridae	Hexapoda	10	22.16	16	29	4.205
Gelastocoridae	Hexapoda	2	21	18	24	4.243
Georissidae	Hexapoda	1	15.43	15.43	15.43	
Gerridae	Hexapoda	2	17.16	16.81	17.5	0.488
Gomphidae	Hexapoda	6	20.64	17.18	25	2.927
Gripopterygidae	Hexapoda	5	18.38	13.9	20.9	2.737
Gyrinidae	Hexapoda	6	21.3	17	25	3.15
Haliplidae	Hexapoda	5	24.58	17.1	36.6	7.235
Hebridae	Hexapoda	2	19.5	16	23	4.95
Heleidae	Hexapoda	1	17.26	17.26	17.26	
Hemicorduliidae	Hexapoda	1	26	26	26	
Heteroceridae	Hexapoda	2	17.35	14.7	20	3.748
Hydraenidae	Hexapoda	10	21.5	15.8	27	3.847
Hydrobiidae	Mollusca	2	16.13	16	16.26	0.184
Hydrobiosidae	Hexapoda	1	19.67	19.67	19.67	
Hydrochidae	Hexapoda	3	25.64	17.5	36.6	9.857
Hydrometridae	Hexapoda	2	22.75	19.5	26	4.596
Hydrophilidae	Hexapoda	30	24.71	14.82	36.6	6.08
Hydropsychidae	Hexapoda	3	23.03	23	23.05	0.029
Hydroptilidae	Hexapoda	16	19.08	13.3	25	3.795
Hygrobiidae	Hexapoda	2	19.57	16	23.14	5.049
Hyriidae	Mollusca	3	18.25	13.46	23.19	4.867
Ilyocryptidae	Crustacea	3	19	16	22	3
Ilyocyprididae	Crustacea	3	21.53	15	25.6	5.714
Laophontidae	Crustacea	2	19.5	16	23	4.95
Leptoceridae	Hexapoda	40	18.52	9.64	36.6	4.868
Leptocytheridae	Hexapoda	2	18	16	20	2.828
Leptophlebiidae	Hexapoda	13	17.81	14.51	20.9	2.189
Lestidae	Hexapoda	9	22.69	12	36.6	8.301
Libellulidae	Hexapoda	7	23.84	15.3	36.6	6.649
Limnadiidae	Crustacea	6	16.8	7.5	22	5.744
Limnichidae	Hexapoda	1	23	23	23	
Limnocytheridae	Crustacea	11	23.14	11	36.6	9.271
Lymnaeidae	Mollusca	2	17.96	17.81	18.11	0.212
Lynceidae	Crustacea	1	36.6	36.6	36.6	
Macrothricidae	Crustacea	13	21.38	14	36.6	6.091
Megapodagrionidae	Hexapoda	2	19.69	18.78	20.6	1.287
Melitidae	Crustacea	1	22	22	22	
Mesoveliidae	Hexapoda	1	23.03	23.03	23.03	
Moinidae	Crustacea	3	20.33	16	24	4.041

Family	Group	n	Mean	Min	Max	s.d.
Muscidae	Hexapoda	11	23.33	17.7	29	3.715
Neoniphargidae	Crustacea	1	20.2	20.2	20.2	
Neotrichidae	Crustacea	3	18.8	16	20.2	2.425
Noteridae	Hexapoda	1	17.3	17.3	17.3	
Notodromadidae	Crustacea	8	20.08	17	23.19	2.697
Notonectidae	Hexapoda	14	23.08	10.87	36.6	7.155
Nymphulinae	Hexapoda	6	20.31	17.11	22.61	1.99
Ochteridae	Hexapoda	1	15.8	15.8	15.8	
Oniscidae	Crustacea	3	24.41	20.2	30	5.044
Oxygastridae	Hexapoda	1	16.38	16.38	16.38	
Palaemonidae	Crustacea	1	23.19	23.19	23.19	
Paramelitidae	Crustacea	2	18.12	15.8	20.43	3.274
Parartemiidae	Crustacea	11	22.24	16.9	30	4.638
Parastacidae	Crustacea	7	21.49	18.3	25	2.591
Parastenocarididae	Crustacea	5	20.71	16	26	4.017
Perthidae	Crustacea	4	20.62	17.8	24.98	3.185
Philopotamidae	Hexapoda	2	20.46	18.1	22.82	3.338
Philosciidae	Crustacea	1	20.6	20.6	20.6	
Planorbidae	Mollusca	7	22.89	18.9	36.6	6.46
Pleidae	Hexapoda	8	18.66	12	23.14	3.526
Pomatiopsidae	Mollusca	9	26.46	21	30.8	3.696
Psychodidae	Hexapoda	6	20.99	14	27	4.654
Ptiliidae	Hexapoda	1	17.48	17.48	17.48	
Pyralidae	Hexapoda	1	23.1	23.1	23.1	
Saldidae	Hexapoda	3	25.93	18	36.6	9.598
Scatopsidae	Hexapoda	2	21.9	19.5	24.3	3.394
Sciomyzidae	Hexapoda	2	26.48	16.37	36.6	14.31
Scirtidae	Hexapoda	3	22.04	18.1	25.6	3.764
Sididae	Crustacea	4	24.3	15	36.6	9.273
Simuliidae	Hexapoda	4	21.25	20.51	23	1.177
Spercheidae	Hexapoda	1	15.02	15.02	15.02	
Sphaeriidae	Mollusca	3	20.11	14.9	25	5.058
Sphaeromatidae	Crustacea	2	21.21	20.43	22	1.11
Staphylinidae	Hexapoda	2	22.34	19.08	25.6	4.61
Stratiomyidae	Hexapoda	1	28.3	28.3	28.3	
Styloniscidae	Crustacea	1	20.2	20.2	20.2	
Sulcaniidae	Crustacea	1	20	20	20	
Synthemistidae	Hexapoda	7	15.79	12.01	18.32	2.557
Syrphidae	Hexapoda	2	22.79	20.6	24.98	3.097
Tabanidae	Hexapoda	1	26	26	26	
Talitridae	Crustacea	1	22.42	22.42	22.42	
Tanyderidae	Hexapoda	2	16.67	15.02	18.32	2.333
Telephlebiidae	Hexapoda	1	19.67	19.67	19.67	
Thamnocephalidae	Crustacea	9	19.22	7.5	25.6	5.085
Thaumeliidae	Hexapoda	1	16.97	16.97	16.97	

Family	Group	n	Mean	Min	Max	s.d.
Tipulidae	Hexapoda	10	19.13	12	28.2	5.564
Trapeziidae	Crustacea	1	16	16	16	
Triopsidae	Crustacea	2	18	15	21	4.243
Veliidae	Hexapoda	4	20.68	10.7	26	6.799

For the eight insect orders investigated, mean MFD temperatures ranged from 18.4°C in the Ephemeroptera (mayflies) and Plecoptera (stoneflies) to 22.2°C in the Coleoptera (beetles) (Figure 6). Mean MFD temperatures were significantly different among the insect orders, with values for the Ephemeroptera and Plecoptera being significantly lower than for the Hemiptera and Coleoptera (ANOVA and Tukeys test, $p < 0.001$).

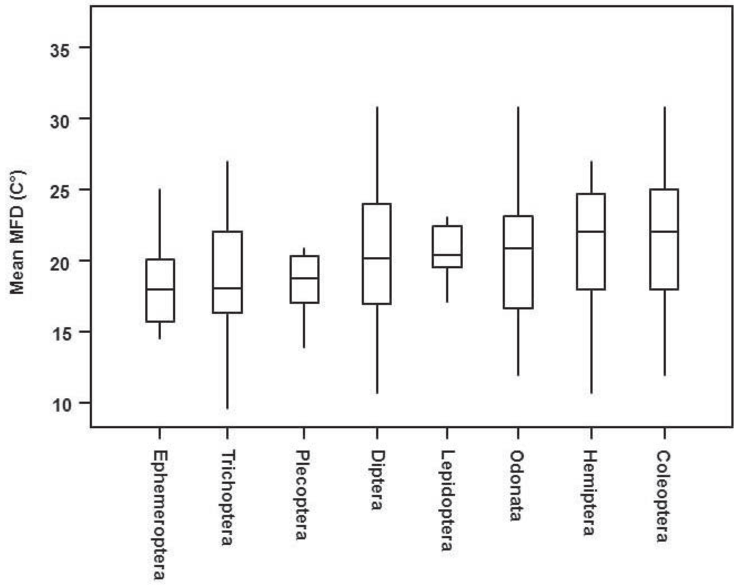

Figure 6. Plot of maximum field distribution temperatures (MFD) selected orders of aquatic insects. Orders have been arranged from left to right according to median MFD values. Boxes span the interquartile range of values, horizontal line indicates the median, vertical whiskers extending beyond the boxes represent minimum and maximum values.

Mean MFD temperatures were estimated at family level for molluscs. For the seven families investigated, mean MFD temperatures ranged from 16.1°C in the family Hydrobiidae to 26.5°C in the family Pomatiopsidae (Figure 7). Although mean MFD temperatures were not significantly different at the 5% level due to small sample sizes (ANOVA, $p = 0.16$), the data do suggest a differential response to temperature amongst molluscan families.

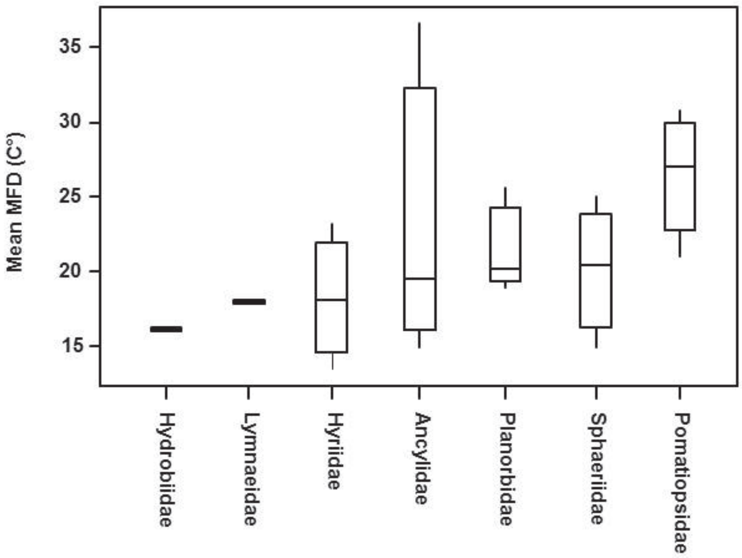

Figure 7. Plot of maximum field distribution temperatures (MFD) for selected families of Mollusca. Families have been arranged from left to right according to median MFD values. Boxes span the interquartile range of values, horizontal line indicates the median, vertical whiskers extending beyond the boxes represent minimum and maximum values.

Mean MFD temperatures were similar across the five major crustacean groups investigated (ANOVA, $p = 0.50$), with values ranging from 21.0-22.5°C (Figure 8).

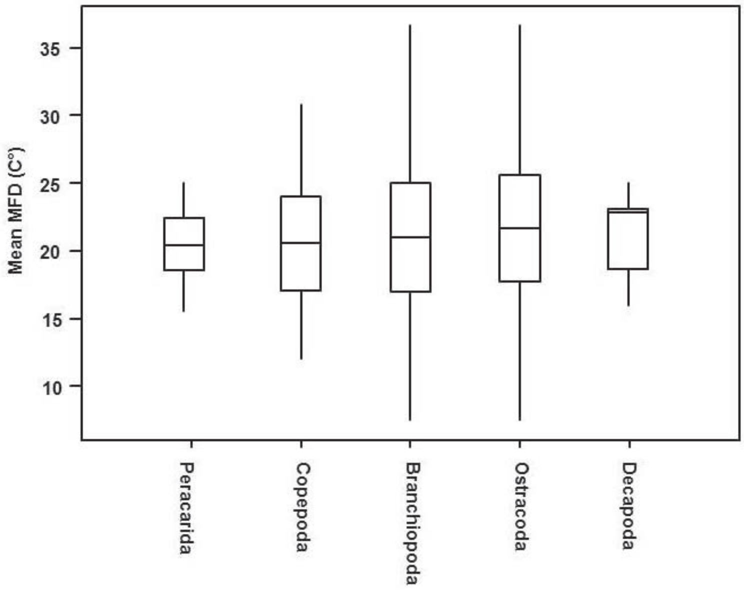

Figure 8. Plot of maximum field distribution temperatures (MFD) for selected crustacean groups. Taxa have been arranged from left to right according to median MFD values. Boxes span the interquartile range of values, horizontal line indicates the median, vertical whiskers extending beyond the boxes represent minimum and maximum values.

3.3.2 Relationship between experimental and field data

In all 16 macroinvertebrate families considered, mean MFD values were always less than their LT_{50} values. There was a relatively weak but statistically significant correlation between LT_{50} and MFD ($r = 0.52$, $p = 0.04$, $n = 16$) (Figure 9).

3.3.3 Scenario testing

When observed versus expected frequencies of 'sensitive', 'tolerant' and 'very tolerant' taxa at five unshaded sites sampled on two occasions were compared, these were found to be significantly different for five of 10 comparisons (Figure 10; Chi-square, $p < 0.05$). On all occasions where frequencies differed significantly (comparisons of A2 /B2 and A4/B4 sampled in both 2006 and 2007 and A5/B5 sampled in 2007), the number of sensitive taxa at unshaded sites was markedly less than at nearby shaded sites.

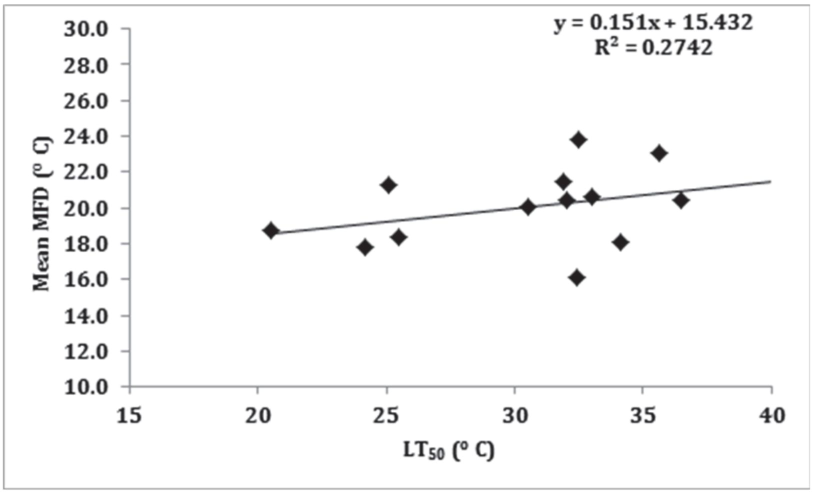

Figure 9. Relationship between mean maximum field distribution (MFD) and experimentally derived lethal temperatures (LT50) for stream invertebrates.

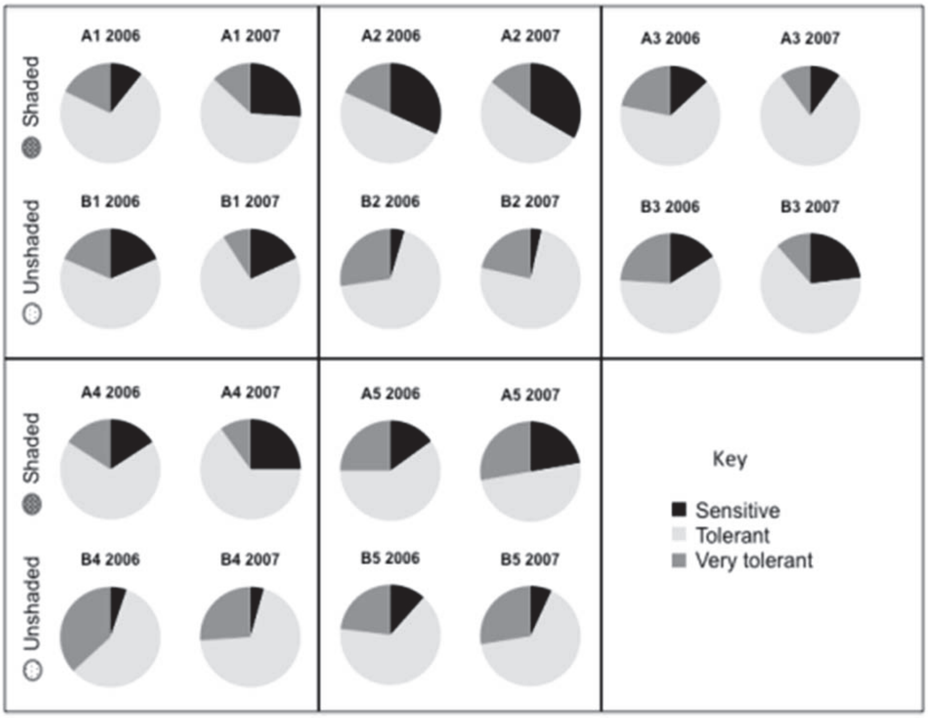

Figure 10. Relative proportions of temperature 'sensitive', 'tolerant' and 'very tolerant' aquatic invertebrates present at paired shaded (A) and unshaded (B) sites in Marbellup Brook (see Figure 5) sampled on two occasions (2006 and 2007).

3.4 Discussion

Additional thermal shifts, due to continued removal of riparian zones or climate change, are expected to result in novel or hybrid ecosystems (*sensu* Hobbs et al., 2009; Walther et al., 2009; Catford et al., 2012) characterized by altered freshwater species assemblages. We predict that the structure of these assemblages will be largely determined by taxonomic differences in thermal tolerance. Elsewhere, elevated temperatures due to climate change have led to movement of species to more cooler regions either at higher latitudes or altitudes (Jacobsen et al., 1997; Davies, 2010). In southwestern Australia, low relief, and surrounding ocean and desert limit the availability of cool water refugia. Consequently, assemblage changes may be characterised by a progressive loss of temperature-sensitive species and the filling of elevated-temperature niches by more tolerant taxa. This has implications for the sustainability of regionally important endemic cool-water species (Horwitz et al. 2008).

Based on the analysis of UTTs presented here, we expect that mayflies may become more restricted in distribution, whereas those species more tolerant of elevated temperatures (e.g. Coleoptera and Odonata) may become more abundant, or increase in geographical range (where migration and recolonisation pathways allow). Although not addressed in this study, sublethal effects of elevated water temperatures may also be important and lead to changes in community structure. These sublethal effects could include changed behavioural responses. For example, for caddisflies with upper thermal limits around 31˚C, Galleep (1977) found a decrease in filtering rate at temperatures above 24˚C and suggested that these larvae were unlikely to thrive and reproduce at higher water temperatures, well below upper thermal tolerance levels.

The correlation, albeit a weak one, between laboratory-derived LT_{50} values and field-derived MFD values suggests that studies of upper thermal tolerances in the laboratory are, to some degree, predictive of the temperature levels at which invertebrates are known to be associated with in the field. Studies comparing acute tolerances and field-collected data are limited; none exist for temperature. The only studies of this kind in Australia are those of Kefford et al. (2004) and Horrigan et al. (2007) who were able to show that at the family level, laboratory-derived and field-derived salinity tolerance levels were related in stream invertebrates. In the light of a limited laboratory-derived database for LT_{50} values, the demonstration of a relationship between LT_{50} and MFD values for stream invertebrates suggests the use of the latter as an interim approach for estimating UTTs of stream invertebrates.

Some studies which have looked at macroinvertebrate assemblages associated with shaded and unshaded stream reaches (e.g. Quinn & Hickey 1990; Quinn et al. 1997; Hawkins et al. 1997; Storey & Cowley 1997; Sponseller et al. 2001) have found reduced macroinvertebrate diversity at unshaded sites. We have demonstrated similar responses by stream invertebrates in southwestern Australia, suggesting that UTT data for stream invertebrates can be used to set biodiversity targets for stream restoration aimed temperature control.

4. USING UPPER THERMAL TOLERANCES TO SET BIODIVERSITY TARGETS FOR RIPARIAN RESTORATION

4.1 Introduction

Modelling studies have demonstrated that planting trees on stream banks can reduce daily maximum water temperatures (Theurer et al., 1985; McBride et al., 1993) with some of this research (e.g. Rutherford et al., 1997; 2004) predicting the extent of cover and length of rehabilitation required to restore or maintain stream temperatures within the thermal tolerance of keystone species. These studies consequently rely on thermal tolerance data for stream invertebrates to set these limits. Although the thermal tolerances of aquatic invertebrates occurring in streams in USA (e.g. De Kozlowski & Bunting,1981; Claussen & Walters, 1982), South Africa (Buchanan et al., 1988) and New Zealand (Quinn et al., 1994) have been determined, data for Australian taxa are limited, and those reported here (Chapter 2) thus represent an important contribution to our knowledge of the thermal tolerances of Australian stream invertebrates.

The thermal testing of the fauna of southwestern Australia (Chapter 2) and the estimation of MFDs (Chapter 3) suggest that 19-21°C as the UTT for a range of ‘sensitive’ freshwater insect taxa. This critical threshold temperature is consistent with that reported elsewhere for a range of temperate species (*e.g.* De Kowzlowski & Bunting, 1981; Quinn et al., 1994; Cox & Rutherford, 2000). In southwestern Australia, this threshold temperature is often exceeded in upland streams flowing through cleared catchments (Rutherford et al., 2004), where the lack of riparian vegetation increases the irradiance into streams (Davies et al., 2004; Davies, 2010) and particularly in reaches running east-west (Davies et al., 2008) where light inputs are maximised (see Osborne & Kovacic, 1993).

The ability to predict characteristics of future ecosystems is crucial for environmental planning and the development of effective climate change adaptation strategies (Davies, 2010). With further thermal shifts in aquatic habitats, we expect novel or hybrid ecosystems (*sensu* Hobbs et al., 2009; Walther et al., 2009; Catford et al., 2012), characterized by altered species assemblages, to develop due to taxonomic differences in thermal optima, tolerance and sensitivity (see Ward & Stanford, 1982 and references therein). The UTT values described in Chapters 2 and 3 now allow for the prediction of likely response to restoration initiatives and consequently the establishment of biodiversity targets for riparian restoration aimed at reducing water temperatures.

This chapter outlines a six-step framework to integrate predictions of water temperatures arising from riparian restoration (e.g as determined using modeling approaches such as SimpSTREAMLINE) with predictions of biodiversity responses based on the differential UTTs of aquatic fauna. In doing so, it will facilitate the establishment of adaptive riparian replanting strategies capable of providing refuges from otherwise increasing water temperatures associated with climate change.

4.2 Approach

Figure 11 shows a framework for integrating predictions of water temperatures arising from riparian restoration with predictions of biodiversity responses based on the differential UTTs of aquatic fauna. The model, SimpSTREAMLINE, developed by Rutherford et al. (1997) and modified by Davies et al. (2004) is a modelling approach that can be used to predict daily temperature fluctuations in streams. When used in combination with digital elevation models for mapping solar radiation and maps showing distribution of stream vegetation, it can identify priority areas in catchments where replanting will increase shading and decrease temperatures.

This modelling approach requires a variety of input data including information on the river channel (depth, velocity and width), solar radiation (latitude, day length and Julian day) and meteorology (cloud cover, air temperature and wind speed) (Figure 11). Using a variety of algorithms (see Davies et al., 2004) the extent of cover and length of rehabilitation required to achieve temperature targets can be predicted (e.g. Rutherford et al., 1997; 2004). Combining these predicted outputs with defined UTTs of aquatic biota (Figure 11) allows iterative (i.e. adaptive) testing of restoration scenarios to identify the required extent and cover of riparian replanting to provide refuges for aquatic biota from high water temperatures. In so doing, it also allows for the establishment of biodiversity targets for riparian restoration aimed at reducing water temperatures (Figure 11).

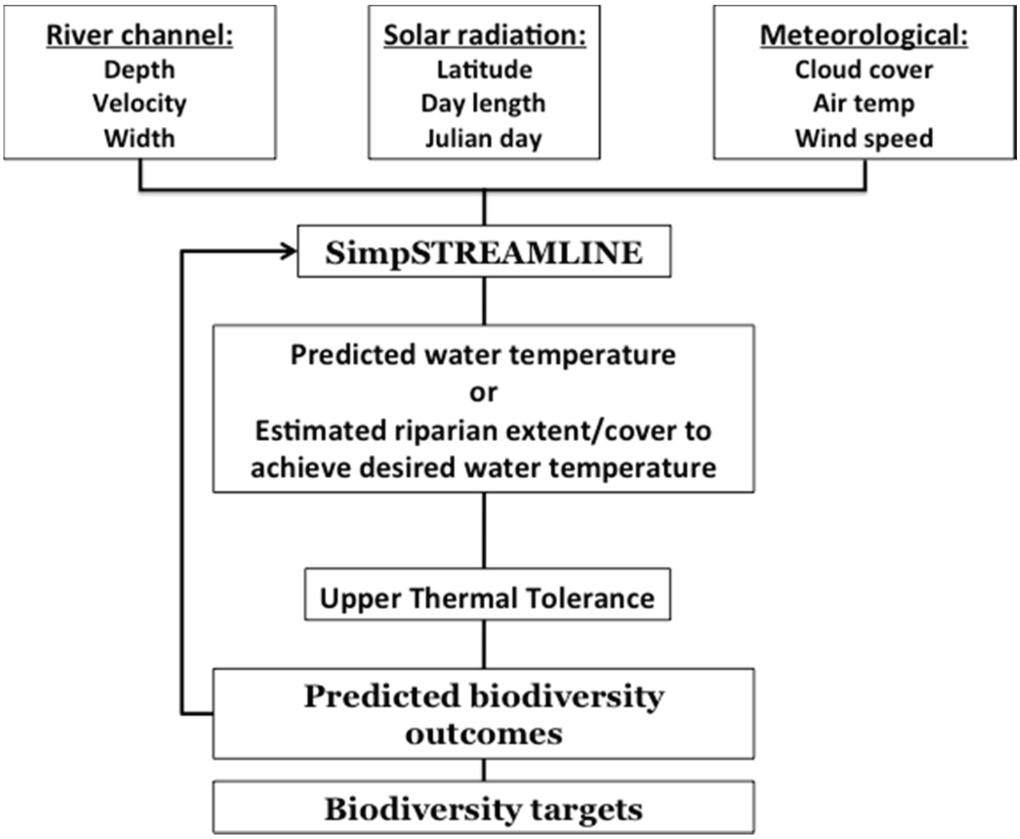

Figure 11. Suggested framework to integrate Upper Thermal Tolerances of aquatic fauna with predictions of water temperature regimes achieved through riparian restoration that will allow for the establishment of biodiversity targets for in situ stream restoration activities.

4.2.1 Five-step approach to set biodiversity targets for riparian restoration

To achieve these outcomes, a five-step process has been developed. This five-step process is outline below.

Step 1 - ***Determine pre and post restoration temperatures*****:** Determined using appropriate tools, such as SimpSTREAMLINE described above.

Step 2 - ***Determine what taxa occur in the catchment, region/bioregion of interest*****:** It is possible that taxa lists are already available for the area of interest based on field sampling. If this is not the case, taxa lists could be compiled based on the existence of aquatic bioregions. For example, five broad aquatic macroinvertebrate regions have been recognised for the state of Victoria (Doeg, 2001). Although aquatic bioregions are yet to be determined for Western Australia, Stewart (2009) did describe two interim aquatic bioregions for the South Coast region. In the absence of taxa lists for the specific sites of interest, taxa known to occur in the defined bioregions could be used to compile a list.

Step 3 - ***Determine the UTTs for taxa present*****:** Upper Thermal Tolerance values for a variety of insect, crustacean and molluscan taxa are reported here and can be used for studies across southern Australia. For other taxa with undefined UTTs, experimental or field derived values (see approaches detailed in Chapters 2 and 3) will be required.

Step 4 - ***Predict faunal assemblage that should occur at both pre and post restoration temperatures*****:** Based on the taxa list for the area of interest, and the pre and post restoration water temperatures, compile lists of taxa that theoretically could occur at the site at these given temperatures. These will be those taxa whose UTTs have not been exceeded. It is possible that the use of taxa lists based on bioregions will list some taxa that can tolerate the temperatures of interest, but are not naturally found at the sites of interest. This will require refinement of the taxa lists to reflect the influence of non-temperature modifiers such as natural variation in distributions, pollution, sedimentation and other disturbances. One way of refining the taxa list would be to compare the assemblages at 'best-available' shaded sites with those predicted.

Step 5 - ***Assess biodiversity outcomes from restoration scenarios (e.g. extent/cover of replanting) and set biodiversity targets*****:** Reduction of water temperatures by restoring riparian shading should lead to an increase in the frequency of temperature-sensitive taxa if the temperature of the restored reaches is within tolerable ranges. If, however, biodiversity outcomes are inadequate (e.g only limited change in proportion of temperature-sensitive taxa) return to step 1 to reevaluate riparian replanting strategy so that instream temperatures are further reduced.

An appropriate way to achieve these five-steps is to establish a Microsoft Access (or similar) database that integrates predictions of water temperatures arising from riparian restoration with predictions of biodiversity responses based on the differential UTTs of aquatic fauna. We have trialed the five-step approach and the use of a Microsoft Access database to successfully set instream biodiversity targets for the Marbellup Brook catchment in southwestern Australia.

4.3 Discussion

A significant feature of the approach described above is that it can be used in subtly, but importantly different ways. First, the approach can be used to define the required extent and cover of riparian replanting needed to achieve predetermined biodiversity outcomes. Aquatic invertebrate assemblages of southern Australian freshwaters are typically of Gondwanan origin, intolerant of elevated water temperatures, and many of these representatives are of conservation significance (e.g. restricted distributions, highly threatened or endemic; Davies, 2010). When restoration is not constrained and maximum temperature reduction is achievable, then the approach can be used to determine the extent of restoration required to achieve a predetermined biodiversity outcome, for example providing appropriate water temperatures for all temperature-sensitive species.

Alternatively, the approach can be used to assess whether planned riparian replanting will result in instream biodiversity outcomes specifically related to water temperature. This may be particularly relevant in situations where the extent of restoration is constrained by for example, available finances, or the length of stream available for restoration. In these circumstances, the five-step approach can be used to determine whether the proposed restoration initiative is likely to provide mitigation against rising water temperatures. In cases where biodiversity outcomes are not significant and resources are limited, these restoration initiatives could be of lower priority. This is not to say that there may be other biodiversity benefits from the planned restoration, such as creation of instream habitat, reduction in nutrient and sediment input or channel stabilisation, that may warrant implementation of the restoration strategy.

REFERENCES

Armstrong, K.N., Storey, A.W. and Davies, P.M. 2005. Effects of catchment clearing and sedimentation on macroinvertebrate communities of cobble habitats in freshwater streams of south Western Australia. *Journal of the Royal Society of Western Australia,* 88, 1-11.

Bates, B.C., Hope, P., Ryan, B., Smith, I. and Charles, S. 2008. Key findings from the Indian ocean Climate Initiative and their impact on policy development in Australia. *Climate Change,* 89, 339-354.

Beatty S.J., Allen, M.A., Lymbery A.J., Storer, T., Morgan, D., and Ryan, T. 2013. *Evaluating small barrier removal to improve refuge connectivity: a global review of barrier decommissioning and a process for southern Australia in a drying climate*. Final Report. NCCARF, Brisbane. 75pp.

Beschta, R.L. 1997. Riparian shade and stream temperature: an alternative perspective. *Rangelands*, 19, 25-28.

Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B. and Sudduth, E. 2005. Synthesizing U.S. River Restoration Efforts. *Science*, 308, 636-637.

Buchanan, J.A., Stewart, B.A. and Davies, B.R. 1988. Thermal acclimation and tolerance to lethal high temperature in the mountain stream amphipod *Paramelita nigroculus* (Barnard). *Comparative Biochemistry and Physiology*, 89A, 425-431.

Bunn, S.E. and Davies, P.M. 1990. Why is the stream fauna of southwestern Australia so impoverished? *Hydrobiologia*, 194, 169–76.

Bunn, S.E. and Davies, P.M., 1992. Community structure of the macroinvertebrate fauna and water chemistry of a salinised river system in south-western Australia. *Hydrobiologia*, 248, 143–60.

Bunn, S.E. and Davies, P.M., 2000. Biological processes in running waters and their implications for the assessment of ecological integrity. *Hydrobiologia*, 422, 61-70.

Bunn, S.E., Davies, P.M. and Mosisch, T.D. 1999. Ecosystem measures of river health and their response to riparian and catchment degradation. *Freshwater Biology*, 41, 333–45.

Calosi, P., Bilton, D.T., Spicer, J.I. and Atfield, A. 2008. Thermal tolerance and geographical range size in the *Agabus brunneus* group of European diving beetles (Coleoptera: Dytiscidae). *Journal of Biogeography*, 35, 295-305.

Calosi, P., Bilton, D.T., Spicer, J.I., Votier, S.C. and Atfield, A. 2010. What determines a species' geographical range? Thermal biology and latitudinal range size relationships in European diving beetles. *Journal of Animal Ecology*, 79, 194-204.

Catford, J.A., Naiman, R.J., Chambers, L.E., Roberts, J., Douglas, M. and Davies, P. 2012. Predicting novel riparian ecosystems in a changing climate. *Ecosystems*. DOI: 10.1007/s10021-012-9566-7

Chester, E.T., Robson, B.J. and Chambers, J.M. 2012. *Draft Technical Report 3 : Anthropogenic refuges for freshwater biodiversity*. Murdoch University, 60 pp.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, 18, 117-143.

Claussen, D.L. 1980. Thermal acclimation in the crayfish, *Oronectes rusticus* and *O. virilis*. *Comparative Biochemistry and Physiology,* 66A, 377-384.

Claussen, D.L. and Walters, L.M. 1982. Thermal acclimation in the freshwater planarians *Dugesia tigrina* and *D. dorotocephala*. *Hydrobiologia*, 94, 231-236.

Cox, T.J. and Rutherford, J.C. 2000. Thermal tolerances of two stream invertebrates exposed to diurnally varying temperature. *New Zealand Journal of Marine and Freshwater Research*, 34, 209-215.

CSIRO. 2007. *Climate Change in Australia*, Technical report 2007. CSIRO Bureau of Meteorology, Melbourne, Victoria, Australia.

CSIRO and Bureau of Meteorology (BOM). 2007. *Climate change in Australia*, Technical report. CSIRO, Australia. 148pp.

CSIRO and Bureau of Meteorology (BOM). 2010. *State of the Climate*. CSIRO & BOM, Australia. 6pp.

CSIRO and Bureau of Meteorology (BOM). 2012. *State of the Climate 2012*. CSIRO & BOM, Australia. 12pp.

Cummings, C.R., Matthews, T.G. and Lester, R.E. 2013. *Novel methods for managing freshwater refuges against climate change in southern Australia: Evaluating the utility of cold-water releases ("shandying") for enhancing the resilience of riverine species,* National Climate Change Adaptation Research Facility, Gold Coast, pp. 73.

Davenport, M.W. and Simons, M.J. 1985. *Waikato coal fired power station investigation: review of aquatic invertebrate monitoring at Huntly and thermal tolerance studies*. Waikato Valley Authority technical report 1985/6, Hamilton, New Zealand.

Davies, P.M., 2010. Climate change implications for river restoration in global biodiversity hotspots. *Restoration Ecology,* 18, 261-268.

Davies, P.M., Bunn, S.E. and Hamilton, S.K. 2008. Primary production in tropical streams and rivers. In: Dudgeon, D. (Ed.). *Tropical Stream Ecology*. Elsevier Inc, Amsterdam. p 23-42.

Davies, P.M., Walshe, T. and Cook, B. 2004. *Managing high in-stream temperatures using riparian vegetation*. River and Riparian Land Management Technical Guideline, Number 7. Land and Water Australia.

De Deckker, P. 1986. What happened to the Australian aquatic biota 18,000 years ago? In: De Deckker, P. and Williams, W.D. (Editors). *Limnology in Australia*. CSIRO Publishing, Melbourne.

De Kowzlowski, S.J., and Bunting, D.L. 1981. A laboratory study of the thermal tolerance of four south-eastern stream insect species (Trichoptera, Ephemeroptera). *Hydrobiologia*, 79, 141-145.

Dunlop, M. and Brown, P.R. 2008. *Implications of climate change for Australia's National Reserve System: A preliminary assessment*. Report to the Department of Climate Change, February 2008. Department of Climate Change, Canberra, Australia. ISBN 978-1-921297-73-1.

Ernst, M.R., Beitnger, T.L. and Stewart, K.W. 1984. Critical thermal maxima of nympha of three Plecoptera species from an Ozark foothill stream. *Freshwater Invertebrate Biology,* 3, 80-85.

Galleep, G.W., 1977. Responses of caddisfly larvae (brachycentrus spp.) to temperature, food availability and current velocity. American Midland naturalist 98: 59-84.

Garten, C.T. and Gentry, J.B. 1976. Thermal tolerance of dragonfly nymphs. II. Comparison of nymphs from control and thermally altered environments. *Physiological Zoology*, 49, 206-213.

Gaufin, A.R. and Hern, S. 1971. Laboratory studies on tolerance of aquatic insects to heated waters. *Journal of Kansas Entomological Society*, 44, 240-245.

Hawkins, C.P., Hogue, J.N., Decker, L.M. and Feminella, J.W. 1997. Channel morphology, water temperature, and assemblage structure of stream invertebrates. *Journal of the North American Benthological Society*, 16, 728-749.

Heiman, D.R. and Knight, A.W. 1972. Upper-lethal temperature relations of nymphs of the stonefly, *Paragnetina media*. *Hydrobiologia,* 39, 479-493.

Hennessy, K., Fitzharris, B. Bates, B.C., Harvey, N. Howden, S.M., Hughes, L., Salinger, J. and Warrick, R. 2007. Australia and New Zealand. Climate Change 2007: Impacts, Adaptation and Vulnerability. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. and Hanson, C.E. (Editors). *Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK, 507-540.

Hobbs, RJ, Higgs, E. and Harris, J.A. 2009. Novel ecosystems: implications for conservation and restoration. *Trends Ecology and Evolution*, 24, 599–605.

Horne, A.J. and Goldman, C.R. 1994. *Limnology*. 2nd Edition. McGraw-Hill Inc.

Horrigan, N., Dunlop, J.E., Kefford, B.J. and Zavahir, F. 2007. Acute toxicity largely reflects the salinity sensitivity of stream macroinvertebrates derived using field distributions. *Marine and Freshwater Research*, 58, 178-186.

Horwitz, P., Bradshaw, D., Hopper, S., Davies, P. Froend, P. and Bradshaw, F. 2008. Hydrological change escalates risk of ecosystem stress in Australia's threatened biodiversity hotspot. *Journal of the Royal Society of Western Australia*, 91, 1–11.

Johnson, S.L. 2004. Factors influencing stream temperatures in small streams: substrate effects and a shading experiment. *Canadian Journal of Fisheries and Aquatic Sciences,* 61, 913-923.

Johnson, S.L. and Jones, J.A. 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences*, 57 (Suppl. 2), 30-39.

Kefford B.J., Papas, P.J., Metzeling, L. and Nugegoda, D. 2004. Do laboratory salinity tolerances of freshwater animals correspond with their field salinity? *Environmental Pollution*, 129, 355-362.

Larson, L.L. and Larson, S.L. 1996. Riparian shade and stream temperature: a perspective. *Rangelands*, 18, 149-152.

Lutterschmidt, W.I. and Hutchison, V.H. 1997. The critical thermal maximum: history and critique. *Canadian Journal of Zoology,* 75, 1561-1574.

Martin, W.J. and Gentry, J.B. 1974. Effect of thermal stress on dragonfly nymphs. In: Gibbons, J.W. and Sharitz, R.R. (Editors). *Thermal Ecology*. Atomic Energy Commission.

McBride, G.B, Cooke, J.G., Cooper, A.B. and Smith, C.M. 1993. *Optimizing the compensation flow requirements of a river impounded for water supply. In: Rivers for our future*. Department of Irrigation and Drainage, Kuala Lumpur, Malaysia, November 1993: pp. 4B/1/1-15.

McKie, B.G., Cranston, P.S. and Pearson, R.G. 2004. Gondwanan mesotherms and cosmopolitan eurytherms: effects of temperature on the development and survival of Australian Chironomidae (Diptera) from tropical and temperate populations. *Marine and Freshwater Research*, 55, 759-768.

Moulton, S.R., Beitinger, T.L., Stewart, K.W. and Currie, R.J. 1993. Upper temperature tolerance of four species of caddisflies (Insecta: Trichoptera). *Journal of Freshwater Ecology*, 8, 193-197.

Nakata, K., Kazuyoshi, H., Hamano, T., Hayashi, K.-I and Kawai, T. 2002. Lethal limits of high temperature for two crayfishes, the native species *Cambaroides japonicus* and the alien species *Pacifastacus leniusculus* in Japan. *Fisheries Science*, 68, 763-767.

Nebeker, A.V. and Lemke, A.E. 1968. Preliminary studies on the tolerance of aquatic insects to heated waters. *Journal of the Kansas Entomological Society*, 41, 413-418.

Norris, R.H., Prosser, I.P., Young, B., Liston, P., Bauer, N., Davies, N., Dyer, F., Linke, S. and Thoms, M. 2001. The assessment of river condition (ARC): an audit of the ecological condition of Australian Rivers. Final Report submitted to the National Land and Water Resources Audit Office.

Osborne, L.L. and Kovacic, D.A. 1993. Riparian vegetated buffer strips in water quality restoration and stream management. *Freshwater Biology*, 29, 243-258.

Poff, N.L, Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E. and Stromberg, J.C. 1997. The natural flow regime: a paradigm for river conservation and restoration. *Bioscience*, 47, 769–84.

Price, P, Lovett, L. and Davies, P. 2008. *National Synthesis of River Restoration Projects: A study undertaken for the National Water Commission within the Water Dependent Ecosystems Theme of the Raising National Water Standards Program*. Canberra, ACT, Australia. ISBN: 978-1-921107-89-4.

Quinn, J.M., Steele, G.L., Hickey, C.W. and Vickers, M.L. 1994. Upper thermal tolerances of twelve New Zealand stream invertebrate species. *New Zealand Journal of Marine and Freshwater Research*, 28, 391-397.

Rutherford, J.C., Blackett, S., Blackett, C., Saito, L and Davies-Colley, R.J. 1997. Predicting the effects of shade on water temperature in small streams. *New Zealand Journal of Marine and Freshwater Research,* 31, 707-721.

Rutherford, J.C., Marsh, N.A., Davies, P.M. and Bunn, S.E. 2004. Effects of patchy shade on stream water temperatures; how quickly do small streams heat and cool? *Marine and Freshwater Research*, 55, 737-748.

Simons, M. 1984. *Species-specific responses of freshwater organisms to elevated water temperatures*. Waikato Valley Authority Technical Publication 29.

Theurer, F. D., Lines, I. and Nelson, T. 1985. Interaction between riparian vegetation, water temperature and salmonid habitat in the Tucannon River. *Water Resources Bulletin*, 21, 53-64.

Tukey, J.W. 1977. *Exploratory Data Analysis*. Addison-Wesley.

Walther G-R., Roques, A., Hulme, P.E., Sykes, M.T., Pyšek, P., Kühn, I., Zobel, M., Bacher, S., Botta-Dukát, Z., Bugmann, H., Czúcz, B., Dauber, J., Hickler, T., Jarosík, V., Kenis, M., Klotz, S., Minchin, D., Moora, M., Nentwig, W., Ott, J., Panov, V.E., Reineking, B., Robinet, C., Semenchenko, V., Solarz, W., Thuiller, W., Vilà, M., Vohland, K. and Settele, J. 2009. Alien species in a warmer world: risks and opportunities. *Trends in Ecology & Evolution*, 24, 686-693.

Ward, J.V. and Stanford, J.A. 1982. Thermal response in the evolutionary ecology of aquatic insects. *Annual Review of Entomology*, 27, 97-117.

Winterbourn, M.J. 1969. Water temperature as a factor limiting the distribution of *Potamopyrgus antipodarum* (Gastropoda-Prosobranchia) in the New Zealand thermal region. *New Zealand Journal of Marine and Freshwater Research*, 3, 453-458.

Murdoch UNIVERSITY

Australian Government
National Water Commission

NCCARF
National
Climate Change Adaptation
Research Facility